Waste Management 58 (2016) 369–375

Contents lists available at ScienceDirect

Waste Management

journal homepage: www.elsevier.com/locate/wasman

Influence of aeration on volatile sulfur compounds (VSCs) and NH_3 emissions during aerobic composting of kitchen waste

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ARTICLE INFO

Article history:
Received 9 May 2016
Revised 22 August 2016
Accepted 23 August 2016
Available online 29 August 2016

Keywords:
Volatile sulfur compounds (VSCs)
NH_3
Aerobic
Kitchen waste
Composting

ABSTRACT

This study investigates the influence of aeration on volatile sulfur compounds (VSCs) and ammonia (NH_3) emissions during kitchen waste composting. Aerobic composting of kitchen waste and cornstalks was conducted at a ratio of 85:15 (wet weight basis) in 60 L reactors for 30 days. The gas emissions were analyzed with force aeration at rates of 0.1 (A1), 0.2 (A2) and 0.3 (A3) L (kg DM min)$^{-1}$, respectively. Results showed that VSCs emission at the low aeration rate (A1) was more significant than that at other two rates (i.e., A2 and A3 treatment), where no considerable emission difference was observed. On the other hand, NH_3 emission reduced as the aeration rate decreased. It is noteworthy that the aeration rate did not significantly affect the compost quality. These results suggest that the aeration rate of 0.2 L (kg DM min)$^{-1}$ may be applied to control VSCs and NH_3 emissions during kitchen waste composting.

1. Introduction

With gradual changes in the structure of energy consumption and improvements of living standards in China, the proportion of kitchen waste to municipal solid waste (MSW), is increasing distinctly (Xu et al., 2011). There were 490 million tones of MSW generated in China in 2014, and more than 40% was kitchen waste (Sheng, 2015; Dong and Qu, 2001). In some areas, kitchen waste accounts for more than 60% due to the high percentage of fruit, vegetable, and food waste (Lin et al., 2011; Wang and Nie, 2001; Wang, 2009). Kitchen waste contains a high proportion of biodegradable organic compounds, and can therefore serve as a renewable resource in MSW. Owing to the characteristics of high moisture content, high content of organic components and salt, and low calorific value (Li et al., 2009; Zhao et al., 2009), it is sustainable to handle kitchen waste as organic waste.

Composting is a biological process that can reduce the volume and mass of kitchen waste. Over the last decades, composting has been widely used for converting organic wastes into relatively stable products for use as fertilizer or for soil amendment (Lin et al., 2008; Nair and Okamitsu, 2010). However, the imbalanced nutrients, such as carbon and nitrogen, high moisture content, and compact structure may discharge by-products, such as malodor and leachate, which can cause secondary environmental pollution, especially in countries with high population density. These factors limit the usage of kitchen waste as mono-substrate in the composting process (Pagans et al., 2006; Domingo and Nadal, 2009).

Odor pollution caused by volatile trace compounds, such as volatile sulfur compounds (VSCs) released from kitchen waste, is a critical public concern, although the potential negative effects of these odors on human health remains debatable (Capelli et al., 2011; Li et al., 2013). Zhang et al. (2012) reported that 43 gaseous compounds were detected during kitchen waste composting, including 6 VSCs, 22 hydrocarbons, 11 aromatic hydrocarbons, and 5 other odorous substances. In their study, the odor intensity was significantly correlated with the concentrations of VSCs emitted during kitchen waste composting ($p < 0.01$), and approximately 55% of the initial total sulfur was lost in the forms of VSCs. VSCs are volatile organic compounds (VOCs) that contain sulfur and reduced sulfur compounds (RSCs), such as hydrogen sulfide (H_2S), methyl mercaptan (MM), dimethyl sulfide (DMS), dimethyl disulfide (DMDS), carbonyl sulfide (OCS), and carbon disulfide (CS_2) (Panetta et al., 2005). The VSCs compounds are characterized by a low detection threshold and strong odor activities, and therefore contribute to odorous pollution even at very low emission concentrations (Yu et al., 2007). Volatile sulfur compounds (VSCs) have been identified as the predominant odorants emitted during organic waste composting (Komilis et al., 2004; Wu et al., 2010). Leachate is an indirect emission source of odor substances and its production rate is between 20 and 40% (based on the wet

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http://dx.doi.org/10.1016/j.wasman.2016.08.022
0956-053X/

weight of compost material) (Schlegelmilch et al., 2005; Eitzer, 1995; Zhang et al., 2012). Ammonia emission is the largest part in all odor substances. Previous studies showed that 9.6–46% of the initial total nitrogen (TN) is lost in the form of NH_3 during composting (Fukumoto et al., 2003; Morand et al., 2005; Jiang et al., 2013; Zhang, 2012).

The aerobic condition during the composting process is a critical factor determining the discharge of unwanted emissions (Haug, 1993; Smet et al., 1999). It is widely accepted that VSCs are released from organic materials during anaerobic and exothermic or incomplete aerobic fermentation processes (Leach et al., 1999; Statheropoulos et al., 2005; Higgins et al., 2006). Komilis et al. (2004) reported that substantial emission of VSCs could occur, even under aerobic conditions. Poor O_2 transfer caused by insufficient aeration is always considered the main reason for odor gas production during MSW composting (Scaglia et al., 2011). Some studies indicated that increased aeration rates can helps to reduce VSC emission during organic waste composting. Rajamäeki et al. (2005) reported that the concentrations of DMDS and H_2S were higher in the composter bin with low aeration (~0.5 L/min) than in that with high aeration (~9 L/min) during kitchen waste composting. Chen et al. (2011) found that maintaining the O_2 concentration higher than 14% in the pile could reduce H_2S production during large-scale sewage sludge composting. However, some studies show that an increased aeration rate was responsible for an increase in NH_3 emission (Guardia et al., 2008; Elwell et al., 2002; Kim et al., 2009; Osada et al., 2000). Li et al. (2008) observed that NH_3 emission at an aeration rate of 0.50 L/min kg VS was higher than that at other aerations such as 0.125, 0.25 and 0.75 L/min kg initial VS, and the aeration rate of 0.25 L (kg VS min)$^{-1}$ VS was considered to be the best for NH_3, H_2S, and MM control during dairy manure composting with rice straw. Shen et al. (2011) found that NH_3 emission at an aeration rate of 0.2 m^3 min^{-1} m^{-3} was higher than that at other aeration rates, such as 0.01 and 0.1 m^3 min^{-1} m^{-3} during aerobic composting of chicken manure and high C/N waste mixture.

Composting aeration rate seems to be a main parameter influencing gaseous emissions (Scaglia et al., 2011; Pagans et al., 2006; Shen et al., 2011; Turan et al., 2007; Gao et al., 2010; Delgado-Rodriguez et al., 2011). However, little research is currently published on combined VSCs and NH_3 control strategies based on aeration rate during kitchen waste composting.

This study aims to investigate the influence of aeration supply on the emission of VSCs and NH_3 during kitchen waste composting. Corn stalks were used as bulking agents to improve the composting conditions and reduce the leachate production. Gaseous emissions and composting temperature, as well as maturity were systematically compared at different aeration rates. Results reported here may provide a unique insight into the methods to control odorous gaseous emissions during kitchen waste composting.

2. Materials and methods

2.1. Raw composting materials

Kitchen waste was collected from a separate collection system in south Beijing. The kitchen waste consisted of, by wet weight, 54% uneaten vegetables, 23% fruit peels, 14% uneaten portions of meals, and 9% leaves. Chopped cornstalk (with a length of 2–3 cm) was used as the bulking agent. Key physicochemical properties of the raw materials are presented in Table 1. A mixture of kitchen waste and cornstalks at a ratio of 85:15 (wet weight basis) were composted. This mixture ratio can effectively avoid leachate production during composting (Zhang et al., 2012).

Table 1
Characteristics of raw materials (average values ± standard deviations from three replicates).

Materials	TOC (%)[a]	TKN (%)[a]	TS (%)[a]	Moisture content (%)	C/N
Kitchen waste	35.1 ± 3.4	2.5 ± 0.5	0.4 ± 0.0	69.6 ± 5.2	16.2
Cornstalks	43.4 ± 2.9	0.8 ± 0.2	0.3 ± 0.0	9.1 ± 0.3	52.9

TOC: total organic carbon; TKN: total Kjeldahl nitrogen; TS: total sulfur.
[a] Based on dry matter (DM).

2.2. Experimental design and methods

VSC and NH_3 emissions during kitchen waste composting were investigated at three different aeration rates (i.e., 0.1, 0.2 and 0.3 L (kg DM min)$^{-1}$, denoted as A1, A2 and A3 treatment, respectively). All treatments were performed in 60 L heat-insulated composting reactors with forced aeration systems for 30 days. Detailed descriptions of these reactors are also available elsewhere (Yang et al., 2013). In brief, these composting reactors were identical with an inner diameter of 0.36 m and a height of 0.6 m. A C-LGX program was used to control the aeration intervals and record the composting pile temperature. Composting piles were manually turned over once a week. Solid samples were collected from three sampling ports and then mixed thoroughly on day 0, 3, 7, 14, 21, and 30 of composting. A certain amount of the solid samples were air-dried and subsequently grounded to pass through a 1 mm sieve.

2.3. Analytic method

Gas samples were collected ten times from a sampling port on top of the reactors throughout the experiment (i.e., on day 0, 3, 5, 7, 11, 15, 20, 25, and 30). An SOC-01 sampling device (Tianjin Dylan Auto Environmental Protection Sci-tech Company, Ltd., China) was used for VSC collection. VSCs were analyzed based on the method previously reported by Zhang et al. (2013) using a Gas Chromatography-Mass Selective Detector (Model 5975N, Agilent Technologies, USA) coupled with an Entech 7890 Preconcentrator (Entech Instruments Inc., CA, USA). NH_3 was trapped by boric acid (2%) and then titrated against 0.5 M H_2SO_4 for quantification. The O_2 and H_2S were analyzed daily using a portable biogas analyzer (Biogas Check, Geotech, UK). The biogas analyzer has two built-in sensors that were used to determine O_2 and H_2S, with a measurement range of 0–25% (±1% volume) and 0–500 ppm (±2.0% full-scale), respectively.

Ground and sieved samples were used to assess the TN, total organic carbon (TOC), total Kjeldahl nitrogen (TKN), and total sulfur (TS). TKN and TOC were determined according to Chinese National Standard (NY 525-2002). TS and TN were determined using an Elementary analyzer (Elementar Analysensysteme Vario MACRO cube, Germany).

Fresh samples were processed for moisture measurement based on a method reported by Thompson et al. (2002). In addition, the fresh compost samples (n = 3) were also mixed with distilled water (1:10 w/w) and then shaken for 0.5 h. The mixture was centrifuged and the supernatant was filtered through a 0.45 mL filter membrane to determine the germination index (GI). It is well established that GI can be used to assess the phytotoxicity of the compost (Sellami et al., 2008). In this study, twenty pakchoi seeds were distributed on filter papers in Petri dishes (10 cm in diameter) and moistened with 10 mL filtered compost supernatant. Three replicate dishes for each sample were incubated at 20 °C for three days. The number of germinating seeds and root lengths

was measured with distilled water as a control. The GI value was calculated by:

$$GI\ (\%) = \frac{[Seed\ germination\ of\ treatment\ (\%)]\ [Rootlength\ of\ treatment]}{[Seed\ germination\ of\ control\ (\%)]\ [Rootlength\ of\ control]}(\%)$$

2.4. Statistical analysis

One Way Analysis of Variance (ANOVA) was performed to compare composting treatments. Multiple comparisons between every two treatments were conducted using the Least Significant Difference test (LSD-t). SPSS (Windows, release 15.0, 2007) was used for all statistical analyses.

3. Results and discussion

3.1. Temperature and oxygen content

Temperature profiles from different aeration rates during composting are shown in Fig. 1a. All A1, A2 and A3 treatments rapidly entered the thermophilic phase on day 3 (greater than 55 °C). After first turning (at day 7), the temperatures rose again. The thermophilic phase lasted for 15, 10 and 8 days in treatments A1, A2 and A3, respectively. They reached the highest temperatures of 74, 69 and 67 °C, respectively, on day 9 of composting. All treatment temperatures met the Chinese standard (Chinese Standard DB11/T 272-2005, 2005), which states that adequate sanitation requires the composting temperature to be higher than 55 °C for 5–7 days. Thus, all aeration treatments were sufficient for pathogen elimination.

Since the easily degradable carbon was exhausted, microbial activity and the decomposition rates of organic matter slowed down (Shen et al., 2011). As a result, the composting entered the curing phase (Fig. 1a). Regular turning and mixing led to the homogenization and aeration of the compost. Thus, an increase in the temperature of all treatments was observed after the compost was turned. The mature compost was emptied from the reactors at day 30 when the temperatures were around 10 °C (i.e., curing stage). The temperature of the A2 and A3 treatments was lower than that of the A1 treatment, probably due to the cooling effect on the composting reactor caused by the higher aeration rates. Nevertheless, ANOVA results showed no significant differences among the three treatments in terms of composting temperature ($P > 0.05$).

Opposite variations were observed between the outlet O_2 concentration and temperature for all three treatments (Fig. 1a and b). As the experiment progressed, O_2 concentrations decreased sharply in the first week, with an increase thereafter when the composting temperature declined gradually. It is noteworthy that the outlet O_2 content of A1 and A2 treatments remained at a relatively low level (less than 8%) for about 11 and 5 days, respectively, while that of the A3 treatment was only one day. Nevertheless, the O_2 content reached the ambient condition at the end of the composting for all treatments. ANOVA revealed a significant difference in outlet O_2 concentration among these three treatments ($P = 0.001$).

3.2. VSC and NH_3 emissions

VSCs detected in all three treatments included H_2S, MM, DMS, CS_2, and DMDS (Fig. 2). These VSC emissions were relatively low at the beginning of composting, reached their peaks after 4–10 days, and then decreased sharply until leveling off after about 15 days. For all the aeration rates, all VSC emissions, except H_2S, reached a maximum on the fifth day after composting started. H_2S emissions reached the first peak values on day 3, 4, and 5 for the treatments of A1, A2, and A3, respectively. After the first turnover (day 7), the H_2S emissions reached a second peak value on day 8 for all treatments. In a sense, H_2S emissions were more susceptible to aeration rates during kitchen waste composting. Taken altogether, VSCs were mainly emitted in the first two weeks, indicating that the control of VSC emission in the first two weeks was important for odor minimization during kitchen waste composting.

Over 90% of VSC emissions occurred within the first 15 days of composting for all treatments. This result is consistent with previous studies where different organic waste materials were composted (Schlegelmilch et al., 2005; Bipro et al., 2011; Zhang et al., 2011a,b; Shao et al., 2014). It is noted that an increase in the aeration rate resulted in a significant decrease in these VSC emissions (Fig. 2).

It is well known that VSCs are mainly released from organic matter during anaerobic or incomplete aerobic fermentation processes. In this study, all detected VSCs were the highest at the lowest aeration rate (0.1 L (kg DM min)$^{-1}$) and decreased with the increase of aeration rate. This result clearly indicates that air supply is important for the control of VSCs emissions, and that the aeration rate of 0.1 L (kg DM min)$^{-1}$ was not enough to maintaining aerobic conditions in the composting reactors. The VSC emissions could be controlled when the aeration rate exceeded 0.2 L (kg DM min)$^{-1}$ due to better aerobic conditions provided in the kitchen waste composting process.

Table 2 shows the overall emissions of different VSCs for all three treatments. The A1 treatment encountered the highest

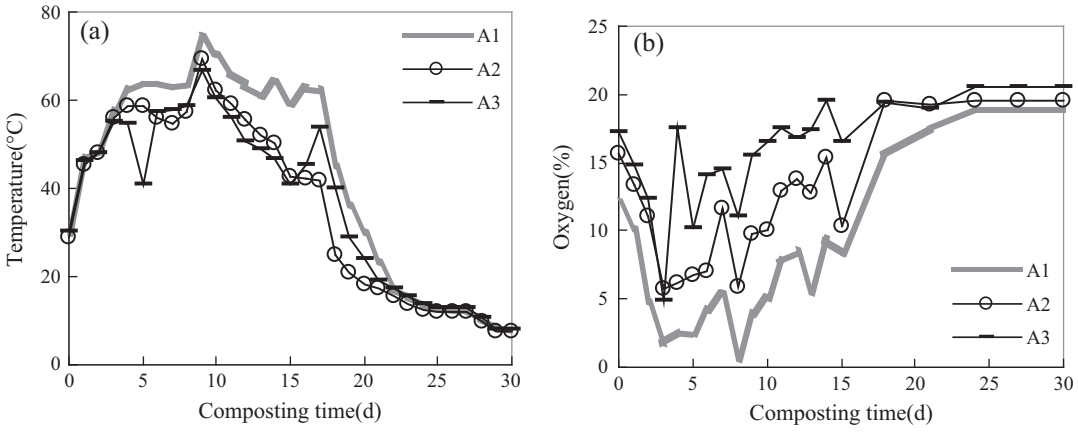

Fig. 1. Change of temperature and O_2 concentrations of all treatments during composing process.

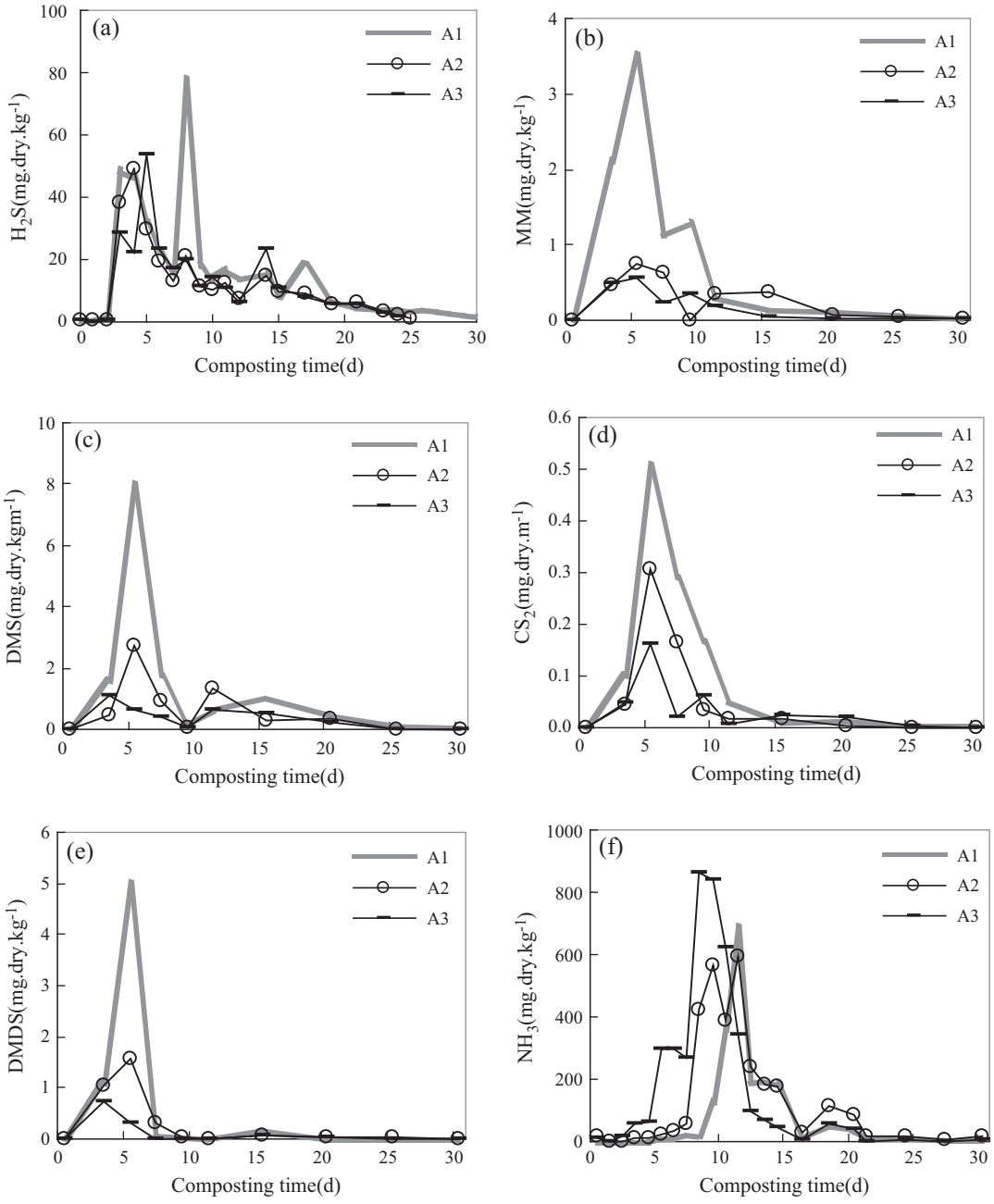

Fig. 2. Emission fluxes for (a) H_2S, (b) MM, (c) DMS, (d) CS_2, (e) DMDS and (f) NH_3.

Table 2
Emissions of different VSCs and NH_3 during composting (mg kg^{-1} DM).

Treatment	H_2S	MM	DMS	CS_2	DMDS	Total VSCs	NH_3
A1	385.5	17.6	33.3	2.2	15.1	453.7	1944.6
A2	301.6	6.5	15.4	1.1	6.7	331.3	3021.3
A3	296.2	4.2	10.2	0.7	3.1	314.4	3995.3

overall VSC emissions, which was 27.0% and 30.7% higher than that in the A2 and A3 treatment, respectively. A similar result was also found in H_2S emissions. Of the three treatments, the A3 treatment produced the largest H_2S emission (Table 2). Of the total VSCs released, H_2S was the most abundant compound, accounting for approximately 85.0, 91.0 and 94.2% of VSCs in the A1, A2 and A3 treatments, respectively. This result is different from those reported previously. Mao et al. (2006) reported that DMS was the major VSC produced during food waste composting. However, Wu et al. (2010) demonstrated that DMDS was the dominant VSC from food waste composting. In a subsequent study, Zhang et al. (2011a,b) reported that MM was a major VSC generated in food waste composting. Such a difference is acceptable, as H_2S emission was not accounted for in these studies. One-way ANOVA showed a significant difference among the three treatments for all VSC cumulative emissions over the whole composting process ($P < 0.05$). However, the LSD-t showed no significant difference between the A2 and A3 treatments, but a significant difference from the A1 treatment. Therefore, VSC emissions during composting might be controlled at an aeration rates higher than 0.2 L (kg DM $min)^{-1}$.

MM, DMS, CS_2, and DMDS showed varying emission patterns in response to the different aeration rates (Fig. 2). Compared to the A1

treatment, the A2 treatment reduced the cumulative emissions of MM and DMS by 63.1% and 76.1%, respectively (Fig. 2b and c). At the same time, the cumulative emissions of CS_2 and DMDS in the treatment of A2 were reduced by 50.0% and 68.2% than in the treatment of A1, respectively (Fig. 2d and e). More considerable reduction in these VSCs emissions was observed as the aeration increased (i.e., the A3 treatment).

NH_3 emissions were relatively low for all three treatments at the beginning of the experiment due to the low ammonia concentration and temperature (Fig. 2f). As the composting continued, the reactor temperature increased significantly, resulting in a considerable increase in the NH_3 emissions for all three treatments. This observation can be attributed to the ammonification of organic nitrogen compounds. NH_3 emissions reached peak values on day 11, 9 and 8 for the treatments A1, A2 and A3, respectively. NH_3 was mainly emitted during the second week, with small amount emitted in the following two weeks of composting. This indicated that the control of NH_3 emissions in the second week is important for odor minimization during kitchen waste composting.

By comparing the cumulative NH_3 emissions among these three treatments (Table 2), it is obvious that a higher aeration rates resulted in more NH_3 emission. One-way ANOVA showed a significant difference among the three treatments for NH_3 cumulative emissions over the whole composting process ($P = 0.001$). The LSD-t showed no significant difference between the A1 and A2 treatments, but a significant difference from the A3 treatment.

In contrast to VSCs, NH_3 is mainly generated via the oxidative degradation of organic materials due to aerobic microorganisms. Biodegradation and ammonification depend on oxygen supply (i.e., aeration rate). Aeration rate is a main parameter of nitrogen loss through ammonia emission. Changes in the temperature of materials during composting showed similar trends with NH_3 changes for all treatments (Fig. 2). Heat generation in the compost is caused by the activity of aerobic microorganisms. As a result, adequate oxygen supply promoted aerobic decomposition of organic matter, which was accompanied by the release of NH_3 and heat. In this study, the degradation of organic materials in the mixture might be relatively slow in the initial stage of composting (i.e., at an aeration rate of less than 0.1 L (kg DM min)$^{-1}$) due to the shortage of oxygen ($O_2 < 5\%$ in the first 10 days) for the aerobic microorganisms, and correspondingly the lowest NH_3 emission occurred.

The aeration rate and pile temperature play important roles in volatilization of VSCs and NH_3 during kitchen waste composting. In this study, statistical analysis suggested that the emission patterns of the five VSCs were significantly negatively correlated with the aeration rate ($p < 0.05$). On the other hand, NH_3 emissions and aeration rate are significantly correlated ($R = 0.729$, $P = 0.000$), meaning that the highest NH_3 emission occurred at the highest aeration rate. Additionally, VSCs and NH_3 emissions were significantly positively correlated with the composting temperature ($p < 0.05$). The statistical analysis also showed that the temperature and aeration rate were significantly negatively correlated for all treatments ($R_{A1} = -0.826$, $R_{A2} = -0.856$, $R_{A3} = -0.834$). Taken together, these findings indicate that selecting appropriate aeration rates while maintaining a suitable compost temperature can reduce VSC and NH_3 emissions during composting, and that the choice of aeration rate is most important.

3.3. Changes in germination rate index and compost maturity

The GI is a significant parameter for evaluating compost phytotoxicity and maturity, and is known to increase with the decomposition of toxic materials in compost, such as short chain volatile fatty acids (mainly acetic acid) (Guo et al., 2012). Cunha et al. (2007) reported that a GI of more than 80% was an indication of a phytotoxic-free medium for plants. Koa et al. (2008) reported that a GI of more than 90% indicated mature compost.

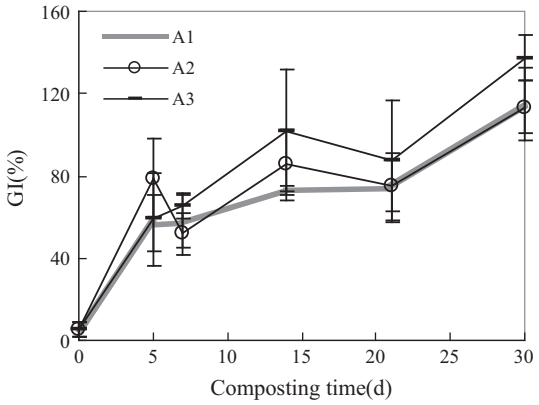

Fig. 3. Changes in germination rate index (GI) of all treatments during composing process.

At the beginning of composting, the GI of each treatment was only about 5% (Fig. 3), indicating the kitchen waste has a high toxic material content. With the progress of the composting, the toxic compounds were rapidly decomposed, and the GI of all three treatments increased. At the end of the composting, the GI in the A1, A2, and A3 treatments reached 114.7%, 113.5% and 137.4%, respectively. Therefore, the conditions in all treatments achieved compost maturity.

3.4. Nitrogen, sulfur and mass balances

Nitrogen is an important nutrient index for compost products. It is well established that during the whole composting process most nitrogen is lost in the form of NH_3 (Szanto et al., 2007; Jiang et al., 2013). In this study, 16–30% of the TN was lost in the form of NH_3 (Table 3). The loss of nitrogen from treatments with the lowest aeration rate (A1) was much less than from the other two treatments, which was probably caused by the lower ammonia removal in A1 (Fig. 2f). In addition to nitrogen, sulfur is also an important substance in organic waste for plant growth in the land applications (Bao et al., 2010). Throughout the composting process, the TS loss from the three treatments was in the ranged 11–17% (Table 3). The A1, A2 and A3 treatments released 11.4, 8.7 and 9.0% VSCs-S as a proportion of the initial TS, indicating that the most sulfur was lost in the form of VSCs. The loss of DM from the treatments of A1, A2 and A3 were 33.1, 31.4 and 33.2%, respectively (Table 3). The loss of DM from A2 was slightly less than from the other two treatments.

Table 3
Nitrogen, sulfur and mass balances of different aeration treatments during composting.

Treatment	Nitrogen balance (%)[a]		Sulfur balance (%)[b]		Mass of DM (kg)		
	NH_3-N	Total N loss[c]	VSCs-S	Total S loss[c]	Initial	Final	Loss[c]
A1	16.1	26.1	11.4	16.2	8.7	5.84	33.1
A2	22.5	34.8	8.7	11.9	9.7	6.64	31.4
A3	29.5	39.4	9.0	11.0	9.5	6.35	33.2

[a] Percentage of initial total nitrogen, dry weight basis.
[b] Percentage of initial total sulfur, dry weight basis.
[c] Total N loss $(\%) = \frac{(\text{TN of raw materials}) - (\text{TN of final compost})}{\text{TN of raw materials}} (\%)$, total S loss and DM loss are calculated analogously to total N loss.

4. Conclusions

Results reported here indicate that high aeration rates could reduce the emission of VSCs during kitchen waste composting. VSCs emissions at a composting aeration rate of 0.1 L (kg DM min)$^{-1}$ were significantly higher than those with aeration rates of 0.2 L (kg DM min)$^{-1}$. Nevertheless, no further reduction was observed when the aeration rate was increased to 0.3 L (kg DM min)$^{-1}$. On the other hand, NH_3 emission was reduced by a decrease in the aeration rate. It is noteworthy that there was no significant difference in ammonia emissions between the composting aeration rates of 0.1 and 0.2 L (kg DM min)$^{-1}$. Regardless of the aeration rate applied, compost mature could be achieved within 30 days with desirable quality. Based on these results, an aeration rate of 0.2 L (kg DM min)$^{-1}$ is suggested for simultaneously controlling VSCs and NH_3 emissions during kitchen waste composting.

Acknowledgements

This study was financially supported by the Beijing Natural Science Foundation (No. 8144050) and the State Key Laboratory of Solid Waste Reuse for Building Materials (SWR-2014-005), This work was also supported by the National Natural Science Foundation of China (project No. 41275161) and the program of National Twelfth Five-Year Science and technology Support (project No. 2012BAD14B01).

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